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Lake Restoration by Sediment Removal

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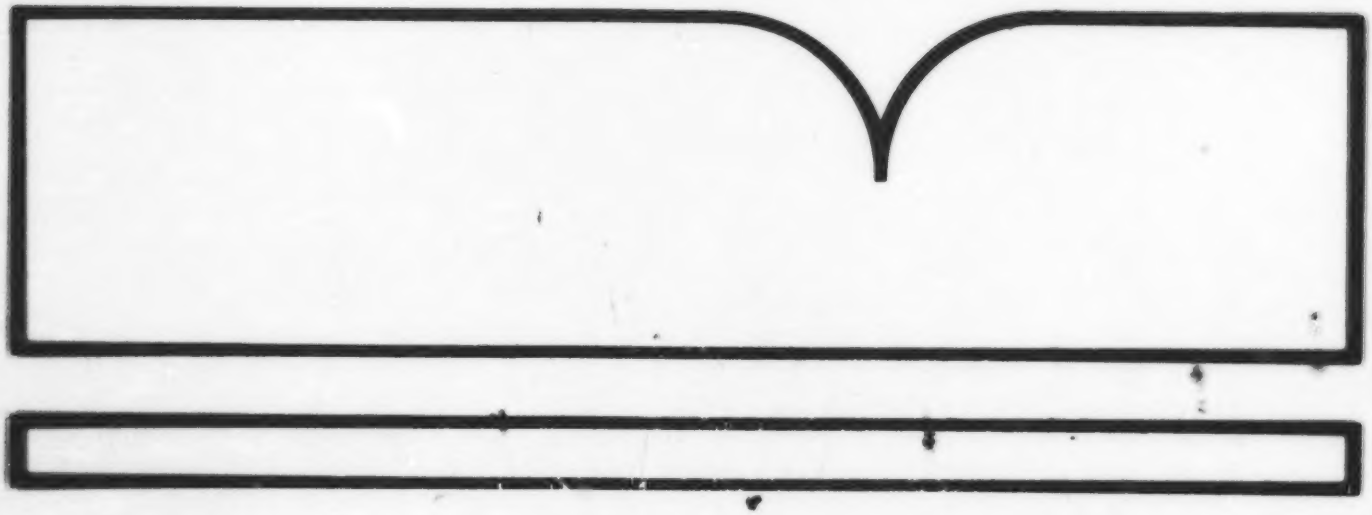

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JOURNAL ARTICLE:

LAKE RESTORATION BY SEDIMENT REMOVAL[1]

Spencer A. Peterson[2]

ABSTRACT: Fresh water lake sediment removal is usually undertaken to deepen a lake and increase its volume to enhance fish production, to remove nutrient rich sediment, to remove toxic or hazardous material, or to reduce the abundance of rooted aquatic plants. Review of more than 60 projects and five case histories reveals that the first three objectives are usually met through sediment removal. Dredging to control aquatic plants has not been well documented. Disadvantages of dredging include cost, temporary phosphorus release from sediment, increased phytoplankton productivity, noise, lake drawdown, temporary reduction in benthic fish food organisms, the potential for toxic material release to the overlying water and potential for environmental degradation at the dredged material disposal site. The technique is recommended for deepening and for long range reduction of phosphorus release from sediment. Sediment removal to control toxic materials is possible with minimal environmental impact when proper equipment is used, but it may more than double the cost. Lack of definitive information about rooted plant regrowth rates in dredged areas prohibits explicit recommendations on sediment removal to control plant growth.
(KEY TERMS: lake restoration; dredging; sediment; eutrophication; phosphorus.)

INTRODUCTION

While control of pollutant inputs should be given high priority in any lake restoration/protection plan, external controls alone may not produce the desired changes (Emery, *et al.*, 1973; Peterson, *et al.*, 1976; Larsen, *et al.*, 1975). Sediment removal might be necessary.

Sediment removal usually includes one or more of four basic objectives: (1) to deepen a lake for improved recreational use, (2) to remove toxic sediment, (3) to reduce nuisance aquatic macrophyte growth, and (4) to prevent or reduce the internal nutrient cycling which may represent a significant fraction of the total nutrient loading (Larsen, *et al.*, 1975; Larsen, *et al.*, 1979; Cooke, *et al.*, 1977; W. H. Funk, 1978, unpublished data, Washington State University).

Until recently there has been little evidence of the effectiveness of sediment removal in meeting these goals. Pierce (1970) reviewed 49 projects and concluded that none provided sufficient data to determine the overall effects of lake dredging. In 1974, Dunst and Associates reported on 50 worldwide lake sediment removal projects, emphasizing the lack of documented effects. This paper updates information on the effectiveness of sediment removal as a lake restoration technique.

PURPOSES OF SEDIMENT REMOVAL

Deepening

Sediment removal to maintain navigation channels has been successfully practiced for years (Herbich, 1975). However, most projects have not addressed the source of the problem and thus require regular maintenance. In small lakes, sediment removal may become necessary when uses such as boating are impaired as a direct result of shoaling. Lake volume must be sufficient to offset seepage and evaporation loss. The U.S. Department of Agriculture (1971) recommends small lake and pond depths ranging from 1.5 m to 4.5 m for various parts of the country to compensate for water loss via these mechanisms. Toubier and Westmacott (1976) report that lakes in colder parts of the United States must be at least 4.5 m deep to avoid winter fish kills. Each of these considerations addresses sediment removal from the standpoint of need to deepen the lake basin. Not surprisingly, Pierce's (1970) survey of upper Midwestern dredging projects showed that all deepening projects were "successful." Who can dispute that when sediment is removed from a lake basin, the lake will be deeper?

Nutrient Removal

Nuisance algal growth and nutrient relationships in lakes are well documented; phosphorus is most frequently cited as the limiting nutrient (Bartsch, 1972; Porcella, *et al.*, 1974; Schindler, 1977). Uptake of nutrients from the water and sediment by plants ultimately results in deposition of organic materials into the lake basin. The total phosphorus content of sediments in selected lakes in North America ranges from 0.5 mg/g to 6.0 mg/g dry wt. Some phosphorus is released during spring and fall lake circulation in dimictic lakes. But, in shallow, polymictic lakes, sedimentary phosphorus release may be more frequent, creating greater nuisance problems with the infusion of nutrients to overlying water during summer recreation periods. This influx of nutrients usually results in abundant, undesirable phytoplankton growth, reducing water transparency, increasing green color, and in severe cases seriously depleting dissolved oxygen causing fish kills. Livingston and Boykin (1962) estimated that sediment regenerated phosphorus

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amounted to approximately 45 percent of the phosphorus loading to Linsley Pond, Connecticut. Welch, *et al.* (1979), estimated the phosphorus from sediments in Long Lake, Washington, to be 200-400 kg yr^{-1} or about 25-50 percent of the external loading. Jones and Bowser (1978), citing a personal communication from Callender (1978, U.S. Geological Survey, Reston, Virginia), reported that sediment phosphorus regeneration in White Lake, Michigan, provided 40 percent of that lake's phosphorus loading. Larsen and Schults (1978) stated that, "the supply of phosphorus from sources within a lake can be several times the supplies from external sources for periods of time on the order of months." Shagawa Lake, Minnesota, experiences internally recycled summer phosphorus pulses of approximately 2000-3000 kg during June, July, and August. This compares to an annual loading from waste water (before advanced waste treatment) of 5000-5500 kg, and current loading of about 1000-1500 kg (D. P. Larsen, 1980, personal communication, U.S. Environmental Protection Agency, Corvallis, Oregon). Thus, internal phosphorus loading to Shagawa Lake before advanced waste treatment (AWT) was about 28-35 percent of the total loading. Following AWT, internal phosphorus loading constituted approximately 66 percent of the total. Continued internal nutrient loading to the lake has been responsible for the slow recovery.

It is evident that internal phosphorus loading to a lake can be significant. Thus, sediment removal should be seriously considered where it might be expected to reduce internal loading.

Toxic Substances Removal

Sediment contaminated with toxic and hazardous materials poses a potential problem for all industrialized countries. In many instances these problems simply have not been recognized. This picture is changing, however, as more thorough and sophisticated lake sediment surveys are conducted (Matsubara, 1979; Horn and Hetling, 1978; Bremer, 1979).

Theoretically, toxic materials in sediment might be controlled by covering or in-place detoxification; however, these procedures are relatively untested. The obvious alternative is to remove the polluted sediment. The question is how, without reintroducing toxicants into the water column and without causing secondary pollution problems at the disposal sites. Turner and Fairweather (1974) stated that "Once a body of water is polluted, the most effective means known to remove bottom sediment pollutants is an *efficient,* hydraulic dredge" (my emphasis). While hydraulic dredges effectively move large volumes of dredged material with minimum energy, resuspension of bottom sediment may be a problem. The key to removing polluted sediments is in minimizing sediment-water interface disruption, containment of polluted dredge materials, and treatment of decant water. Several mechanisms for attaining these goals have been described and are discussed in this paper under sediment removal methods.

Rooted Macrophyte Removal

Rooted aquatic plants in the littoral zone of lakes often interfere directly with recreational uses, which may be sufficient justification for removing the plant beds. Dredging is one approach. However, due to cost and convenience differences, harvesting and chemical treatments are more common.

While direct use impairment problems may justify the removal of some macrophytes from lakes, there is increasing literature on the effects of macrophytes on internal nutrient cycling. Their role in this process, with its effect on algal dynamics, may be an even more important reason to control macrophytes by removing them from a lake. Wetzel (1975) indicates that most of the organic matter in small lakes may be derived from their littoral zone. Reimold (1972) showed that salt marsh plants (Spartina) act as pumps, extracting sediment nutrients, translocating them to stems and leaves, and excreting them to the surrounding water on each successive tidal cycle.

There is considerable evidence that, in the active growth phase, healthy fresh water aquatic macrophytes do not excrete large quantities of nutrients to the surrounding water. However, they do tend to concentrate sediment supplied nutrients in their tissues. Some loss occurs through plant part sloughing, but most nutrients are recycled to the lake when plants fruit and during senescence and plant decay (Lie, 1979; Welch, *et al.*, 1979; Barko and Smart, 1980). Barko and Smart (1979) estimated that in-lake mobilization of phosphorus by *Myriophyllum* in Lake Wingra, Wisconsin, might amount to approximately 60 percent of the annual external phosphorus loading. Welch, *et al.* (1979), reported similar findings in Long Lake, Washington.

While macrophyte removal may be incidental to the objectives of many lake deepening and nutrient control projects, removal of these plants may be far more important to reducing in-lake nutrient cycling than had been suspected. Current evidence indicates that any long range restoration project concerned with in-lake nutrient controls will need to focus on both macrophyte and sediment (Carignan and Kalff, 1980; Barko, 1980, personal communication, U.S. Army Corps of Engineers, Vicksburg, Mississippi).

CONSIDERATIONS FOR SEDIMENT REMOVAL

Currently there is no commonly accepted problem assessment procedure, remedial action plan, preferred piece of equipment, or even agreement about when, why, or how to remove sediment from a fresh water lake. But, there are several factors to consider: (1) problem sources, (2) sediment characterization, (3) sediment removal depth, (4) associated environmental problems, (5) sediment removal methods, (6) sediment disposal area, and (7) most suitable lake conditions.

Clearcut, definitive statements about these factors are not always possible. This may be due to lack of information, individual lake circumstances, the geographic and socioeconomic environment, and the project time frame.

Determination of Problem Sources

Although observations of lake use impairments may be the catalyst for action, they are inadequate to provide restoration plans. Quantitative data are needed to identify causes of

perceived problems. A common, useful approach is the determination of nutrient budgets which correlate well with algal and associated problems. Reckhow (1979) reviewed a number of nutrient mass balance approaches and concluded that several satisfactorily predict trophic state for a given set of lakes. However, Vollenweider's (1975) loading criteria have been most generally used due to ease of application in assessing the susceptibility of a lake to altered phosphorus loading.

Most balances focus on methods to account for surface and subsurface nutrient inputs and outputs. They do not account directly for internal nutrient recycling, but can be used to estimate the relative importance of internal nutrient supplies. The internal gain or loss of nutrients can be calculated as the difference between the algebraic sum of the inputs and outputs of the lake and the change in nutrient concentration in the lake over the same time frame. Figure 1 illustrates the potential importance of internal nutrient cycling in shallow lakes and indicates how nutrient washout model predictions of lake recovery might vary from measured in-lake results. Reckhow (1980) has summarized methods for using hydraulic and nutrient budgets to assess lake problems.

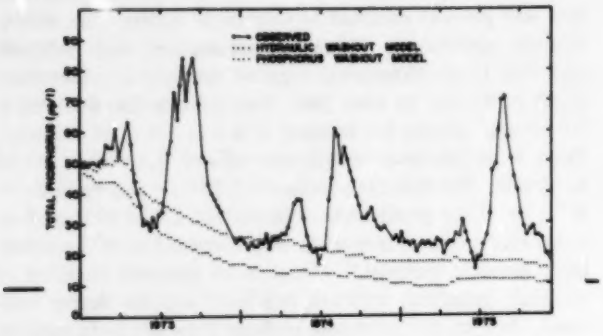

Figure 1. Comparison of Hydraulic and Phosphorus Washout Models with Observations in Shagawa Lake, 1973-1975. Total phosphorus concentrations are the averages for the entire lake (updated from Larsen, *et al.*, 1975).

Characterization of Sediment

If sediment removal appears to be a reasonable lake restoration option, it may be necessary to refine the sediment information base to determine physical characteristics such as sediment depth and distribution, particle size and organic content, and bulk density. Chemically it would be important to know if the sediments contain toxicants, what the nutrient content is, and the rate of nutrient release from the sediment. In the case of nutrients, estimates may be made "directly" via time series analysis using two different approaches: (1) *in situ* isolation of portions of the lake bottom and overlying water suspected of contributing nutrients, or (2) incubation of sediment samples in the laboratory under simulated lake conditions. In either case, sediment nutrient release is measured as change in nutrient content of the overlying water with time. Similar rationale might be used to assess sedimented toxic materials.

One objective of sediment removal for nutrient control purposes might be to skim the upper nutrient rich layer of sediment revealing a strata containing a less readily exchangeable nutrient form (Figure 2). Characterization of pore water and solid phase sediment profiles from representative areas of the lake will permit the development of maps to locate the horizontal and vertical distribution of the relatively nutrient poor strata. For phosphorus limitation planning, these maps should be based on the component phosphorus contents of the sediments as proposed by Williams, *et al.* (1976). Methods for measuring these components are described by Sommers, *et al.* (1972), and by Williams and Mayer (1972). The potential rate of nutrient release can be calculated from sediment characterization data coupled with time series temperature and oxygen profiles.

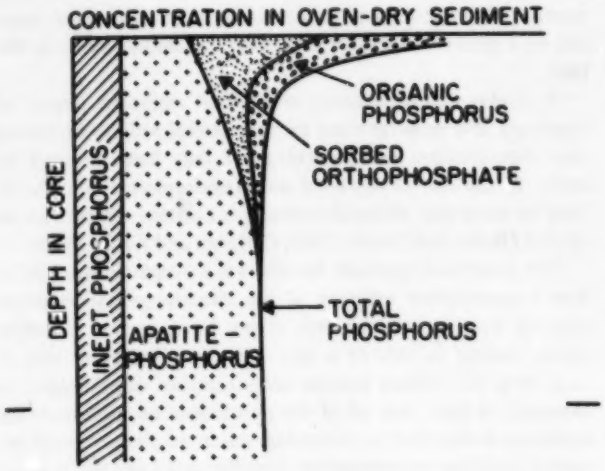

Figure 2. Distribution of Phosphorus in a Sediment Profile from Lake Erie (Williams and Mayer, 1972) Hypothetically Depicted and Illustrating the Diagenesis of Mineral Phosphate from Amorphos to Crystalline Form. Percentage of each form is given by area field (modified from Jones and Bowser, 1978).

Determining sedimentation rates of organic detritus and nutrient elements is important in estimating sediment accumulation rates in the lake basin, and how soon the benefits of sediment removal might be lost if the rate remains high. A common estimation technique involves determining the ragweed pollen (*Ambrosia*) horizon in sediments and measuring the depth of accumulation above that known horizon (Bortleson and Lee, 1972; Craig, 1972). Another method uses the isotopes ^{210}Pb (Koide, *et al.*, 1973) and ^{137}Cs (Pennington, *et al.*, 1973).

Jones and Bowser (1978) point out that pore fluid chemistry, solid phase chemistry, and sedimentation rate may be used to calculate the fluxes of solutes (thus nutrients) into and from the sediments. Close examination of sediment data in conjunction with physical chemical data should help the lake manager determine the significance of internal nutrient release and thus the advisability of removing sediment to control

internal cycling. For example, if one has determined from the literature, *in situ*, or laboratory studies that the phosphorus release rate from anaerobic sediments is on the order of 5 to 10 mg m^{-2} day^{-1}, and that 50 percent of the lake basin in question is intermittently anaerobic for approximately 20 days during the summer, then the potential phosphorus loading to the lake from this source would be: 5 to 10 x 0.50 x 20 = 50 to 100 mg m^{-2} $summer^{-1}$. If the overlying water column was 10 m thick the resultant phosphorus loading to the water might amount to 5 to 10 mg m^{-3} $summer^{-1}$. A comparison of the above phosphorus release rates with those of surface input to the lake and with those of deeper sediment strata will be useful in making a decision about sediment removal. For example, if it could be demonstrated that deeper sediment released phosphorus at a much slower rate, say 2.5 to 5 mg m^{-2} day^{-1}, the loading to the overlying water column from the strata might be reduced to approximately 2.5 to 5 mg m^{-3} $summer^{-1}$, or about a 50 percent reduction. This could translate to a similar magnitude reduction for chlorophyll *a* in the lake.

A similar approach could be used to assess the impact of vegetated and nonvegetated aerobic littoral sediments, taking into consideration the percentage of lake basin covered by each. In the case of vegetated areas (macrophytes) one should keep in mind that seasonal variation in nutrient release may be critical (Barko and Smart, 1980; Carignan and Kalff, 1980).

The analytical approach described is a simplistic example of how a quantitative estimate of the effectiveness of sediment removal for nutrient control might be compared to other major loading sources to a lake. It also provides an idea of how long the effects (minus sedimentation rates) might be expected to last. Not all of the phosphorus released from the sediment is recycled to overlying water, so the potential impact is probably overestimated. Stauffer and Lee (1973), however, have shown that significant amounts of hypolimnetically released phosphorus do reach the epilimnion in shallow lakes as the thermocline is eroded by strong winds. By applying Stauffer and Lee's mixing model approach, or Stauffer's (1980b) "LAKETRANS" model which predicts thermal diffusivities and solute transport in stratified lakes, the precision of impact predictions might be increased. A thorough discussion of internal nutrient loading in lakes and methods for assessing their significance was completed recently by Stauffer (1980a).

Determination of Sediment Removal Depth

To deepen a lake for sailing, power boating, and associated activities, the requirements are relatively straightforward. However, projects to control internal nutrient cycling may focus on open water area sediments or littoral sediments and associated macrophyte beds where removal requirements are not clearly defined.

Lake Trummen, Sweden, provides the best documented case history of sediment removal to control internal nutrient cycling. Sediment removal depth was determined by methods similar to the above example. Sediment was characterized chemically and physically, horizontally and vertically. Digerfeldt (1971) as referenced by Bjork (1972) determined that lake sedimentation rates increased dramatically during the period from 1940 to 1965. About 40 cm of loose FeS-colored (black) fine sediment was deposited during that time. Aerobic and anaerobic release rates of $PO_4^{=}$–P and NH_4^{+}–N from the black sediment layer and the underlying brown layer were shown to be markedly different. These differences were used to develop the plan to remove the upper 40 cm of sediment. Project results are presented as case history No. 1 in this paper.

Stefan and Hanson (1979) and Stefan and Ford (1975) used a lake temperature model to predict the lake depth necessary to prevent summer destratification. Their model inputs air temperature, dew point temperature, wind direction, solar radiation, and wind speed at 12-hour time step intervals, and considers lake morphology, to produce an output of temperature prediction with depth. The approach used by Stefan and Hanson is similar to that developed by Stauffer and Lee (1973) which shows how windpower erodes the thermocline of northern temperate lakes. The major difference is that Stefan and Hanson used their model output to predict dredging depth required to maintain a stable summer stratification, thus preventing enriched hypolimnetic waters from mixing into the epilimnetic zone.

The Stefan-Hanson model assumes stable summer stratification will prevent enriched hypolimnetic waters from mixing into the epilimnion. Using that assumption, they indicated that Hall Lake (Minnesota) required dredging to a maximum depth of 8.0 m. In other data, they indicate that dredging to 8.0 m may require the removal of 4.0 to 6.0 m of sediment. There is no reference to sediment volume, however, it would be sizable. The Hall Lake surface is 2.25 km^2 and mean depth is 2.1 m. Lake profile data indicate that 2.0 to 10.0 m of accumulated sediment is overlain with about 0.5 m of "boundary layer limnetic material." There is no apparent chemical or physical distinction between this layer and the deeper sediment. In fact, sediment core analyses show relatively uniform phosphorus concentration from the surface to 8.5 m (737 to 1412 mg kg^{-1} for 37 samples, mean of 1097). However, there is no indication of phosphorus release rates for sediment from different depths. It is possible that rates from deeper sediment might be less than that of the surface sediment. Thus, reduced nutrient release from deeper sediment may be sufficient to minimize the adverse impact of nutrients on the overlying water, even though stratification was not stable (Bengtsson, *et al*, 1975). If so, it may be possible to remove less sediment at less cost and achieve similar results. Therefore, it seems advisable to examine phosphorus release rates for sediment from various depths before adopting a lake temperature modeling approach to determine dredging depth.

Where sediment removal is contemplated to control nuisance macrophyte growths and associated nutrient cycling, it is necessary to know what depth will prevent the nuisance. Guidelines for this variable are even less definitive than those for controlling nutrient cycling from limnetic zones.

Varied factors contribute to macrophyte growth, e.g., sediment type, nutrient availability, temperature, and light level. The Wisconsin Department of Natural Resources uses light level (determined by Secchi disc) as an indicator of nuisance

macrophyte growth. The average summer water clarity is determined by periodic Secchi disc measurements. The macrophyte growth estimate is determined from a graphic display of maximum plant growth depth (m) as a function of Secchi disc (m). The straight line relationship is described by:

$$Y = 0.83 + 1.22X$$

where:

Y = maximum depth of plant growth in meters, and

X = average summer water clarity (Secchi disc) in meters.

The r^2 for this relationship is 0.53, with N equal to 55. Dunst (1980a) reports that the relationship was based on work by Belonger (1969) and Modlin (1970). Dunst indicates that dredging depth does not always need to exceed the predicted Y-value to effectively control rooted plants (1980, personal communication, Wisconsin Department of Natural Resources, Madison). Simply increasing the water depth can cause speciation changes and in Wisconsin, nuisance surface growths are uncommon where depth exceeds three meters. Also, removing soft organic sediment to a hard sand or clay bottom usually reduces problem plant growth. Unfortunately, truly predictive methodologies are unavailable. It should be noted that dredging for macrophyte control is not synonymous with macrophyte eradication. Any macrophyte dredging plan must consider preservation of fish spawning and nursery areas, waterfowl feeding areas, and other wildlife habitats.

Environmental Problems Associated with Sediment Removal

Peterson (1979) identified potential environmental problems associated with sediment removal. In-lake problems relate to sediment resuspension and the liberation of nutrients with resuspended fine sediment. Phosphorus is of particular concern because of its high concentration in interstitial waters and its affinity for fine particulates. Dredge agitation and wind action tend to move the disturbed nutrient laden sediment into the lake's euphotic zone, producing the potential for algal blooms. Churchill, *et al.* (1975b), reported increased phosphorus concentration in Lake Herman, South Dakota, coincident with dredging but they found no increased algal production, presumably due to the high turbidity. Dunst found that algal increases in Lilly Lake, Wisconsin, was a short term phenomenon associated with active dredging; however, there were no significant adverse effects (1980, personal communication, Wisconsin Department of Natural Resources, Madison).

A potential problem associated with finely divided sediment in the water column is oxygen depletion. If the sediment is highly organic, the particles quickly become bacteria coated. The extensive surface area of these particles may permit rapid decomposition and possible oxygen depletion. Lake dredging might promote oxygen depletion since the organic content of the sediment can approach 80 percent on a dry weight basis (Wetzel, 1975). To my knowledge there is no documentation of this problem in lake dredging projects, but the potential exists.

Liberation of toxic substances is another problem associated with resuspended sediment. Although dredging specifically to remove toxic substances from lakes is relatively uncommon, the practice may increase (Matsubara, 1979; Sakakibara and Hayashi, 1979; Bremer, 1979). The concern for toxics liberation is associated with fine particle size sediment. Murakami and Takeishi (1977) have shown that when dredging in PCB (polychlorinated biphenyl) contaminated sediments, up to 99.7 percent of the PCB materials adhere to particles less than 74 μ in diameter. This could be particularly perplexing in fresh water dredging since particle settling times in fresh water generally are greater than required for brackish or marine waters.

A relatively common concern with lake dredging projects involves destruction of the benthic community. If the lake basin is dredged completely, two to three years may be required to reestablish the benthic fauna (Carline and Brynildson, 1977). However, if portions of the bottom are left, reestablishment may be almost immediate (Andersson, *et al.*, 1975) or within two years (Crumpton and Wilbur, 1974). In any case, the effect on the benthic community appears to be of relatively short duration compared to the longer term benefits derived from sediment removal. The same appears to be true for game fish populations (Carline and Brynildson, 1977; Spitler, 1973).

Sediment Removal Methods

The drawdown/excavate method of sediment removal is summarized by Born, *et al.* (1973). Its applicability is limited to lakes with control structures or those where pumping rates above water inflow rates are possible at reasonable cost.

Dredging is a more common approach to sediment removal. Pierce (1970) described hydraulic dredging equipment and provided guidance on the engineering considerations of dredge selection, i.e., dredge size, pumping rate, pumping distances, etc. Barnard (1978) and Peterson (1979) have described various grab, bucket, and clam shell dredges, hydraulic cutterhead dredges, and specialized dredges to minimize secondary water quality impacts.

The conventional hydraulic cutterhead dredge (Turner and Fairweather, 1974) is most common. Hydraulic dredges have numerous advantages over grab or bucket types. They have almost continuous operating cycles, removing large volumes in a short time. Production rates in excess of 1,000 m^{-3} hr^{-1} are not uncommon. These dredges can span virtually any horizontal reach of water, due to their floating operations platform, and can be "environmentally cleaner" than grabs and buckets, due to their closed cycle operating system. However, because they move large volumes, they may create large turbidity plumes, resuspending sedimented nutrients and hazardous materials. The turbidity around hydraulic cutterheads increases exponentially with the thickness of the cut, rate of swing, and cutter rotation rate (Barnard, 1978). However, Barnard indicated that it is possible to increase dredge

production to the maximum *without* generating excessive turbidity by optimizing the three dredge functions. Minimizing turbidity will depend as much on the skill and experience of the operator as on the texture and particle size of the sediment and general operating conditions.

Where minimizing sediment resuspension is critical to a project (toxic substances removal), it may be necessary to employ specialized equipment. Equipment types and capabilities, described by Barnard (1978) and Peterson (1979), vary from modified grabs and innovative cutterhead design, the newest being the disc bottom cutterhead (Breebot brochure), to air-driven pumps. A number of specialized variations have been used successfully in Japan (Peterson and Randolph, 1977, 1978, 1979). Suda (1979) compared the suspended solids concentration around a conventional cutterhead dredge to that found around specialized Japanese dredges. Results indicated a 10-fold reduction in suspended solids concentration around the specialized dredges. However, no comparisons of production rates were made.

Recently, scientists at the U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi, tested the "pneuma" (air driven pump) dredge system. Although the system's ability to minimize resuspension of sediments was not discussed, McNair indicated that the system was inefficient because of high energy requirements relative to dredge material production (1980, personal communication, U.S. Army Corps of Engineers, Vicksburg, Mississippi). This may explain bid cost differentials between hydraulic and specialized dredging equipment (Peterson, 1979). Thus it is important to weigh the need to minimize sediment resuspension against added cost in any dredging project.

Sediment Disposal Area

The major nonlake impact of sediment removal centers around the disposal site. High population densities make it difficult to locate suitable sites. Another complicating factor is that the indiscriminate filling of low lying areas is no longer permitted under Federal Law (Section 404, Public Law 92-500). Permits are required for virtually all projects where the lakes and adjacent wetland proposed for filling exceeds 4.0 ha (10 acres).

In upland disposal areas (where Federal fill permits are not required) diking is common. A major problem with upland disposal is dike failure (Calhoun, 1979, personal communication, U.S. Army Corps of Engineers, Vicksburg, Mississippi). Another problem is underdesign of disposal area capacity. This situation becomes evident only as the project progresses. Disposal of fresh water sediments may be particularly troublesome because of their slow settling rates (Wechler and Cogley, 1977). As disposal areas fill, ponding depth is reduced and their capacity to retain suspended solids declines. Disposal area suspended solids retentative design must consider ponding capacity at project completion, not some intermediate point. Palermo, *et al.* (1978), summarized technical information and research projects which can assist with proper design, construction, and maintenance of dredged material disposal areas. When standards for discharge from disposal areas cannot be met, it may be necessary to treat the discharge as discussed by Barnard and Hand (1978). Other information addressing the management of disposal areas has been reported by Brannon (1978); Chen, *et al.*, 1978; Gambrell, *et al.*, 1978; and Lunz, *et al.*, 1978.

Lake Conditions Suitable for Sediment Removal

A lake with a filled basin might become a prime candidate for sediment removal. The amount of sediment removed will depend not only on the amount obstructing use of the lake, but also on the availability of suitable disposal sites. Lack of sites need not be a deterrent to sediment removal since there are productive use options for dredged sediment (Lunz, *et al.*, 1978; Walsh and Malkasian, 1978; Spaine, *et al.*, 1978). At Nutting Lake, Massachusetts, approximately 275,000 m^3 of sediment was to be removed and sold for \$1.00 m^{-3} reducing dredging cost to \$0.45 m^{-3} (R. Manfredonia, 1980, personal communication, USEPA, Boston, Massachusetts). This not only solves the disposal problem, but significantly reduces the overall project cost.

Due to the high cost of removing sediment from lakes, \$1.00 to \$15.70 m^{-3} as reported by Peterson (1979), it would seem that lake size might be a major limiting factor for dredging. However, surface area of dredged lakes ranges from 2 ha to over 1,050 ha, and volume removed ranges from a few hundred m^3 to over 7 million m^3 (Peterson, 1979). While technically there is no reason a large lake cannot be dredged, economics will dictate. Unfortunately, most dredging projects have not been scrutinized closely from the economic standpoint.

Current information indicates that lakes with enriched surface sediment, relative to underlying sediment, would benefit from dredging (Andersson, *et al.*, 1975; Bengtsson, *et al.*, 1975). Lake Trummen, Sweden, showed marked improvement after dredging. Similar short term results were observed when 25-50 cm of organic sediment were removed from Steinmetz Lake, New York (Snow, *et al.*, 1980). This project differed significantly from Lake Trummen since clean sand replaced almost all of the organic sediment removed (case history No 4, this paper).

Sediment removal will not be cost effective where high sedimentation rates prevail. Removal projects should have reasonable assurance of extended longevity. Generally, the smaller the ratio of watershed area to lake surface area, the greater the assurance of a lasting effect. Watershed management practices should be encouraged, however, massive programs to protect small lakes may be counterproductive. Uttormark, *et al.* (1974), note that where ratios of watershed area to lake surface area exceed 10 to 1, significant nutrient reduction from the watershed may be impractical.

Generally, small lakes with organically rich sediment, low sedimentation rates, and long hydraulic residence times are the most feasible candidates for sediment removal.

CASE HISTORIES OF SEDIMENT REMOVAL

Sixty-four preimplementation, on-going, or completed sediment removal projects in the United States were summarized by Peterson (1981). Summary information included the objective of specific projects, pre- and post-treatment chemical, physical, and biological characteristics of the lakes, sediment removal method, volume of sediment removed, the percentage of the basin dredged, the cost, the duration of effectiveness, and side effects noted as a result of sediment removal. (The 14-page summary is available from the author.)

The following case histories are presented because: (1) they are examples of different sediment removal approaches, (2) they have different objectives, (3) they illustrate strengths and weaknesses of the technique, and (4) their results have been relatively well documented.

1. Lake Trummen, Sweden

Lake Trummen, Sweden, is one of the most thoroughly documented dredging projects in the world, with both pre- and post-treatment information. This project represents an attempt to reduce internal nutrient cycling by skimming off a thin nutrient rich sediment layer.

The lake has an area of 400 ha and, prior to restoration, a maximum depth of 2 m and a mean depth of 1.1 m. By the early 1960s, domestic waste discharge had so deteriorated the lake that the citizens of Växjo considered filling the basin. Diversion of the major pollutant sources had little effect on the lake. Sediment removal implementation in 1970 was preceded by two years of study and planning. In 1970, one-half meter of finely divided organic sediment was dredged uniformly from the main basin (Bjork, 1974). In 1971, another half meter of sediment was removed, bringing the total volume to about 400,000 m^3. This deepened the lake by approximately 40 cm. Dredged material was partially disposed of in three macrophyte infested, diked-off bays. One large bay, about one-third of the lake area and overgrown with macrophytes, was preserved as a wildlife habitat. Dredged material not placed in diked embayments was pumped to upland diked ponds where return flows were treated with aluminum sulfate to reduce phosphorus concentration in dredge slurry ($\approx$1 mg P $liter^{-1}$ to about 30 μg $liter^{-1}$). Dried dredge material was sold as top soil for about \$2 m^{-3} (U.S.). Phosphorus concentration in the lake prior to sediment removal was 600 μg P $liter^{-1}$. Since the dredging project, concentrations occasionally reach 70 to 110 μg P $liter^{-1}$ (Bengtsson, *et al.*, 1975).

Bengtsson, *et al.* (1975), indicated that phosphorus and nitrogen decreased drastically (Figure 3) and that the role of the sediment in recycling nutrients was minimized. The total orthophosphate as well as the Kjeldahl-nitrogen levels have remained significantly lower than before or during the dredging program.

Biologically, the Shannon phytoplankton diversity index increased from 1.6 in 1968 to 3.0 in 1973 (Cronberg, *et al.*, 1975). Secchi disc transparency increased from 23 to 75 cm over the same period. Prerestoration mean annual phytoplankton productivity was 370 g C m^{-2} (1968-1969), and 225 g C m^{-2} (1972-1973). More than 60 percent of the annual phytoplankton production in 1972-1973 was attributed to algae less than 10 μm in size. Blue-green algal biomass was drastically reduced and some species disappeared, notably *Oscillatoria agardhii.*

Before dredging in Lake Trummen, oligochaetes and chironomids dominated the benthic fauna (Andersson, *et al.*, 1975). A year after dredging tubificid oligochaetes and chironomids became much more numerous. Total number of benthic organisms changed little, probably because of chironomid larvae mobility and the fact that the dominant species swarmed all summer, recolonizing newly dredged areas.

The results at Lake Trummen are encouraging. Born (1979) indicated that the lake is not only a usable recreational resource, but that the lake ecosystem has been restored. Data in Figure 3 indicate that water quality has been remarkably good for four years following dredging compared to prior years. Jorgensen (1980), however, recently stated that, "a deterioration of the water quality has been recorded during the last 2-3 years (1976-1978), caused by the phosphorus input in the mixtures of rain and waste water discharged to the lake during heavy rainfalls." No data were presented to determine what the nature of the deteriorated water quality might be.

2. Lake Herman, South Dakota

Lake Herman, South Dakota, has a surface area of 546 ha, maximum depth of 2.4 m, and mean depth of 1.7 m. The lake has a 145 km^2 agricultural watershed. Row crop and small grain farming practices beginning in the early 1900s have created deposits of about 2 m of silt over the entire lake basin. In 1970, 1971, and 1972, a 4.2 ha area of one bay was dredged. This project is an example of a marginally successful, limited scale hydraulic dredging operation to reduce availability of nutrients in a shallow, intermittently destratified lake.

About 48,000 m^3 of silt were removed from the dredge site, deepening it from 1.7 m to approximately 3.4 m (Figure 4). Dredged material was deposited in a 3.4 ha area adjacent to the lake (Churchill, *et al.*, 1975a). At the end of the dredging project in 1972, drying had reduced the volume of the material about 50 percent. No coagulants were added to the dredge slurry.

Orthophosphorus concentrations in the lake increased dramatically from approximately 0.13 mg P $liter^{-1}$ to $>$ 0.56 mg P $liter^{-1}$ shortly after dredging began (Brashier, *et al.*, 1973). Phosphorus concentrations in the southeast bay paralleled the main lake but appeared to lag by several days (Figure 5). Despite the rapid increase in phosphorus, there was no major increase in phytoplankton productivity because the lake was primarily nitrogen limited. Orthophosphorus concentrations in Lake Herman prior to dredging were about 0.30 mg $liter^{-1}$. During dredging (summer 1970, 1971, and 1972) they were almost uniformly 0.17 to 0.50 mg $liter^{-1}$, while after dredging (summer 1973, 1974, and 1975) they ranged from 0.10 to 0.50 mg $liter^{-1}$ with a tendency toward the lower end of the range (Churchill, *et al.*, 1975b). Churchill and Associates were reluctant to conclude that dredging increased phosphorus in Lake Herman. However, they did indicate that "there were no other noticeable environmental changes that

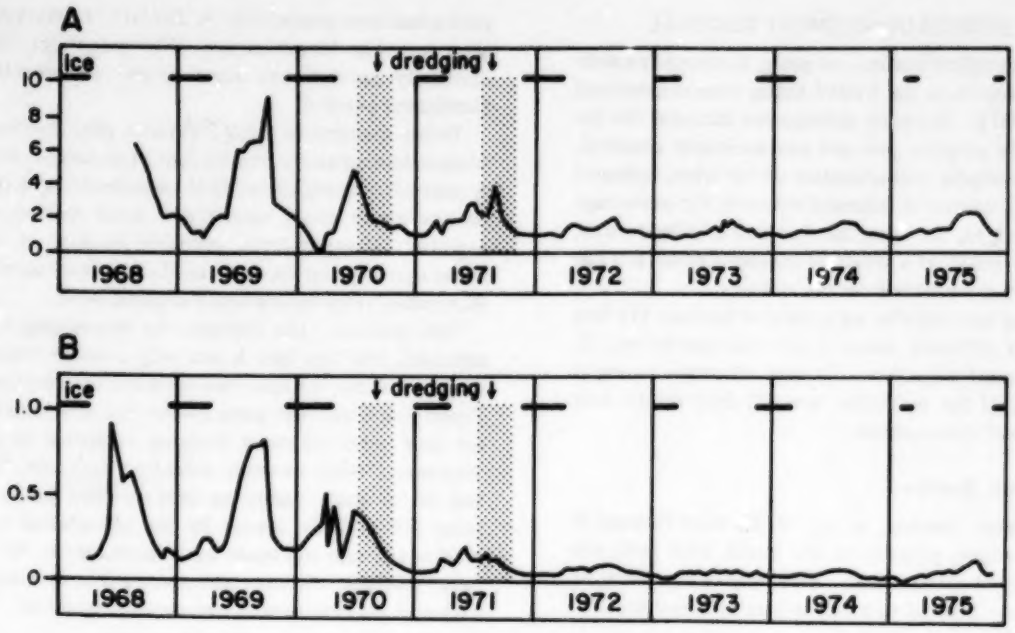

Figure 3. Lake Trummen. Organic nitrogen (A, mg N liter^{-1}) and total phosphorus (B, mg P liter^{-1}) in surface water 1968-1975. Data from W. Ripl (modified from Bjork, 1978).

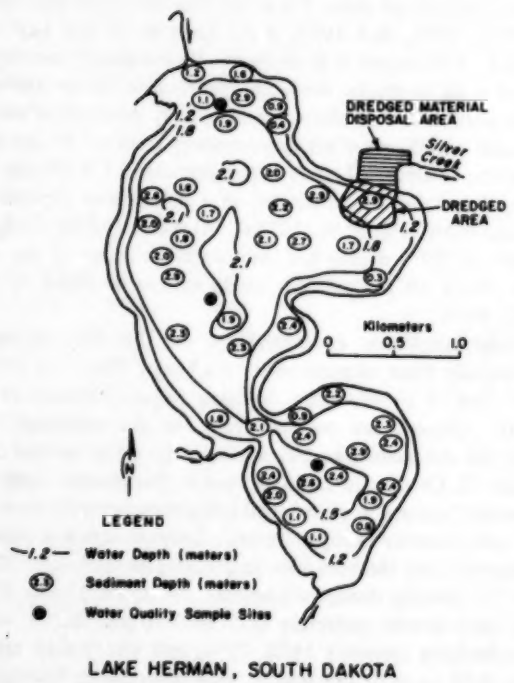

Figure 4. Lake Herman, South Dakota (redrawn from Churchill, *et al.*, 1975a).

could readily account for this dramatic increase in phosphates." While no apparent changes in phytoplankton numbers resulted from the increased phosphorus, that may not have been true if the lake had not been normally nitrogen limited. If the increases in phosphorus were actually attributable to dredging one might conclude that this particular operation was inefficient in limiting sediment resuspension. The project demonstrated that nutrient laden silt could be removed from a small area of the lake by hydraulic dredging, thereby reducing the amount available at that site for internal cycling. However, a question remains regarding the total lake effects, when such limited dredging is done.

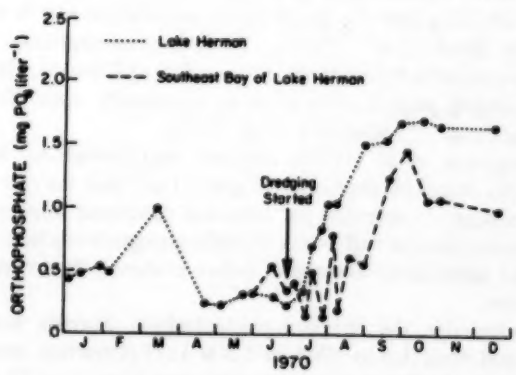

Figure 5. Orthophosphate (X 0.33 = P) in Lake Herman, South Dakota (modified from Brashier, *et al.*, 1973).

3. Wisconsin Spring Ponds

The Wisconsin Spring Ponds (Sunshine and Krause) Project represents another well documented lake dredging project. The dredging was done to deepen and increase pond areas to improve fish production. Although control of aquatic macrophytes was incidental to the program, this was an important plus since Carline and Brynildson (1977) feel that aquatic macrophytes play perhaps the most important role in determining the filling rate in spring ponds. Presumably this conclusion has implications for other types of small closed basin lakes such as Lilly Lake, Wisconsin (Wis. Dept. Nat. Resour., 1969).

Sunshine Spring Pond before dredging had an area of 0.36 ha, a mean depth of 0.46 m, a maximum depth of 1.22 m, and a volume of 1,725 m^3. After dredging these variables were 0.38 ha, 1.68 m, 3.66 m, and 6,910 m^3, respectively. Sediment removal over the entire basin represents the worst possible short run environmental impact. It disrupts the benthic invertebrate population and removes most macrophytes. In Sunshine Pond the predredging mean annual density of benthos was 5,500 organisms m^{-2}. Immediately after dredging the number was reduced to 84 m^{-2}. After five years the density of benthic invertebrates had increased to 16,000 m^{-2}. Results for Krause Pond were similar. However, prior to dredging most of the invertebrates were chironomid larvae; five years later they were primarily oligocheates which are not an important trout food.

Carline and Brynildson (1977) indicated that after four years the important fish food organisms, such as amphipods and chironomid larvae, were still reestablishing. This had a temporary, but statistically insignificant, effect on fish production. Fish growth rates declined sharply in 1971 during the dredging operation, but by 1975 had returned to predredging levels.

Prior to dredging, *Chara* (a nonrooted, nonvascular aquatic macrophyte) covered 60 percent of the Sunshine Pond basin. Dredging removed essentially all of the plant material. After five years *Chara* was reestablished over 28 percent of the pond. However, its biomass was only about 10 percent of the predredging level.

Lake dredging often concerns fisheries managers because of suspected adverse impacts on fish, such as destruction of fish food organisms. Experience in the spring ponds indicates that benthic invertebrate reductions had only limited, short term impact on the fishery. Carline and Brynildson (1977) indicated that four to five years after dredging the average density and biomass of fish were substantially greater than before.

4. Steinmetz Lake, New York

This case history is presented to illustrate a unique whole lake treatment technique: complete drawdown, bulldozer removal of 25 to 50 cm of organic sediment, and replacement of the sediment with clean quarry sand. Macrophyte control and beach improvement were the objectives.

Steinmetz Lake, in Schenectady, New York, was once a quarry (Snow, *et al.*, 1980). The city acquired the lake and surrounding property in 1935. In 1936, the lake was drained and some deep holes filled to improve swimming. The city maintains it for recreational use.

The lake is 1.2 ha in area, with a mean depth of 1.5 m and a maximum depth of 2.1 m. The 28-40 ha naturally vegetated watershed is a public park (Snow, *et al.*, 1980). Day use of swimming and park facilities declined rapidly from 17,000 users in 1971 to 6,000 in 1975. The decline was attributed to excessive weeds and turbidity caused by excessive nutrient loading/algal production from storm water drains and surface runoff, lawn fertilizing in the watershed, and the in-lake macrophyte growth/death/decay cycle (Snow, *et al.*, 1980).

Restoration, beginning in 1978, included sediment removal, diversion of storm water drains, and construction of surface runoff interceptor drains. Bulldozing 2026 m^3 of sediment required more time than anticipated (six weeks) because equipment kept sinking in the soft sediment (Snow, *et al.*, 1980). Backfilling with clean sand was much easier with 1843 m^3 being placed in eight days. This technique creates no net changes of lake bathymetry, but produced a new, clean substrate.

The short term results of the Steinmetz Project are dramatic. Snow, *et al.* (1980), and Bloomfield (1980, personal communication, New York State Department of Environmental Conservation, Albany) indicate the average summer Secchi disc readings improved from 1976 and 1977 at 1.25 and 1.33 m, respectively, to post-treatment levels of 1.47 and 2.20 m (maximum depth of the lake) in 1978 and 1979. In 1976 and 1977 average summer chlorophyll *a* levels were 10.4 and 5.8 μg $liter^{-1}$, respectively. Post-treatment levels have been reduced to 0.1 μg $liter^{-1}$ in 1978 and 1.8 μg $liter^{-1}$ in 1979. Aquatic macrophyte biomass, primarily *Potamogeton crispus*, at 30-50 g wet weight m^{-2} before sediment removal, was reduced to virtually zero. After sediment removal *P. crispus* was observed growing at irregular intervals, in straight lines at a density of 2 to 20 plants m^{-2}. Closer study revealed that plants were growing where tracked vehicles had forced sediment up through the sand cover during the sand spreading operation. Apparently, the extruded organic sediment provided a rooting media for *P. crispus*.

Snow and Associates (1980) rated the project successful. In 1977, public use was virtually zero. In 1978, 2,050 people used the beach and in 1979 the number increased to 3,285.

5. Lilly Lake, Wisconsin

Lilly Lake, in southeastern Wisconsin (Kenosha County), has a 37 ha area and a 155 ha watershed. The lake suffered from severe shoaling and had a mean and maximum depth of 1.4 m and 1.8 m, respectively. Dunst (1980a) found the basin filled with 10.7 m of sediment containing masses of nondecomposed plant material.

A Wisconsin Department of Natural Resources (DNR) report (1969) identified the main problem at Lilly Lake as shallow depth and overly abundant plant growth. There was no indication of plant density. However, it was noted that vegetation was removed mechanically for several years in efforts to maintain boating and swimming areas. The DNR report identified *P. amplifolius* as the dominant plant and notes that in spring 1966 total phosphate concentration at 1 m was 0.26 μg

liter^{-1} (0.09 μg P liter^{-1}). Dissolved phosphate concentration was 0.04 μg liter^{-1} (0.01 μg P liter^{-1}).

The fish population was dominated by slow growing yellow perch. Chemical eradication and restocking with largemouth bass, sunfish, and northern pike had been generally unsuccessful (Wisconsin DNR, 1969). Winter fish kills were common. It was concluded that deeper water was required to manage for fish production. The DNR recommended that at least 10 percent of the lake be dredged to 6.0 m; there is no indication how that specific depth and area were determined.

In 1976 Wisconsin received a grant from the U.S. Environmental Protection Agency to restore Lilly Lake. The State's plan called for dredging about 665 x 10^3 m^3 of sediment, deepening the lake to about 6 m. The objective was to deepen the lake enough to alleviate plant growth and winter fish kill problems (Wisconsin DNR, 1975). Most of the dredge material was to be disposed of in a nearby gravel pit.

Initial indications were that the sediment would flow to a suction dredge intake and that cutterhead swing could be eliminated. However, when dredging began in July 1978, it was discovered that the plant material in the sediment formed a tight matrix which resisted flow. From July to October approximately 360 x 10^3 m^3 of sediment were pumped from the lake (Dunst and Beauheim, 1979). During the 1978 dredging period, in-lake 5-day BOD was about 1-2 mg O_2 liter^{-1} greater than in 1977. Turbidity increased from 1 to 3 formazin units. Ammonia concentration increased from approximately 0.01 mg liter^{-1} in 1977 to nearly 5.5 mg liter^{-1} in October 1978. Although predredging records of particulate phosphorus concentrations were incomplete a steady increase was apparent during dredging. The 1977 chlorophyll *a* levels were consistently 2.5 to 3.0 μg liter^{-1}. In 1978 chlorophyll *a* reached approximately 27 μg liter^{-1} immediately after dredging began and then dropped to 12 to 18 μg liter^{-1} for the rest of the dredging season. Dunst and Beauheim (1979) reported that while productivity had averaged about 200 mg C m^{-3} day^{-1} during the summer of 1976 and 1977, it increased to 750 mg C m^{-3} day^{-1} in 1978.

Dredging began again in May 1979 and was completed by September (Dunst, 1980a). Sediment removal was slightly more than 680 x 10^3 m^3 and the lake now has a maximum depth of 6.5 m. Additional Lilly Lake dredging data are available (Dunst, 1981). In-lake macrophyte biomass of 200 to 300 g dry wt m^{-2} in 1977 was reduced to virtually zero when dredging ceased in September 1979 (R. C. Dunst, 1980, personal communication, Wisconsin Department of Natural Resources, Madison). Water quality has remained good and local sponsors are pleased with the project results (Dunst and Beauheim, 1980).

COSTS OF SEDIMENT REMOVAL

Sediment removal costs vary widely. Controlling factors include type and quantity of sediment; removal method, i.e., bulldozer, mechanical hydraulic or special purpose dredge; location and distance to disposal site; method of transport; and ultimate use of dredged material. Even within the same geographic region, using the same dredging equipment, costs vary. Carline and Brynildson (1977) indicated that cost per cubic meter of sediment removal to increase the volume of the Wisconsin spring ponds ranged from $0.52 to $2.67. National cost differentials are much greater (0.22 to 13.93 m^{-3}). The Army Corps of Engineers has shown a similar broad range in sediment removal costs on a national basis (Saucier, 1976).

Generally, when disposal is not a problem, per unit volume costs for sediment removal is inversely related to the total volume removed. Contaminated material, requiring treatment and perhaps upland containment, may increase cost by three to five times (Peterson, 1979). On the other hand, sale of "desirable" dredged material may help offset removal costs. Sediment from Nutting Lake, Massachusetts brought $1.40 m^{-3} (R. Manfredonia, 1980, personal communication, USEPA, Boston, Massachusetts). Some sediment from Lake Trummen, Sweden, sold for nearly $2.00 m^{-3}. Innovative disposal may reduce the expense of sediment removal, but closer scrutiny of cost is needed before reliable generalizations are possible.

SUMMARY

1. Sediment is removed from a lake: (1) to deepen it for boating, fishing, swimming, and related recreational activities; (2) to reduce the effects of toxicants; (3) to remove and/or reduce nuisance plant growth; and (4) to prevent or reduce the internal sediment/water recycling of nutrients. The anticipated results of treatment are improved dissolved oxygen concentrations; less sediment water nutrient exchange, accompanied by reduced algal biomasses; increased clarity; and reduced macrophyte nuisances.

2. The technique is effective for deepening: when a lake has shoaled extensively there may be no other practical restoration method. Sediment removal is accompanied by nutrient removal which undoubtedly contributes to overall lake quality improvement in many projects.

The duration of deepening effectiveness will depend largely on lake sedimentation rates. The Lake Trummen experience of skimming nutrient rich surface sediments to control internal nutrient cycling was dramatically successful for approximately six years. Signs of deteriorating water quality since then were associated with external nutrient loading. However, water quality has remained better than it was prior to dredging.

Macrophyte control by sediment removal is usually attempted by deepening; effectiveness has not been well documented. Sediment texture, nutrient content, and water temperature may be as important as light level in determining if plants reinvade a dredged area.

3. The reduction of toxic problems in lakes by sediment removal is technically feasible. The practice is highly sophisticated in Japan. It requires special dredging equipment, confined disposal areas, and careful treatment of dredged material return flow waters. Costs are usually two to four times more than for removing uncontaminated sediment.

4. Unsuccessful sediment removal projects are usually associated with extremely limited sediment removal (simply too

small a scale to be effective), an inefficient dredging operation (poor production of solids in the dredge slurry, excessive downtime, extreme turbidity problems, or inadequate disposal areas), or no attempt to control external loading of pollutants to the lake.

5. Any project considered for sediment removal should be preceded by an investigation of nutrient and sediment mass balances. Mass balance calculations provide information on the relative proportion of pollutants derived from various sources, thus establishing of a mechanism for focusing on critical compartments. Accurate sedimentation rates permit the calculation of refilling rates, thus providing a means of comparing the cost effectiveness of sediment removal with other restoration methods.

6. Numerous environmental concerns are associated with sediment removal. Most appear to be of short duration and thus limited significance when compared to the longer range effectiveness of the treatment. This may not be true when toxic substances are involved. If sediment removal is poorly managed, effects of disposal may be longer lasting and will directly relate to disposal method and dredged material quality. Filling of wetlands (no longer common) results in permanent destruction of that habitat. Enrichment of agricultural land with nontoxic dredge material may provide long-term benefits. Imaginative project plans can reduce the impact of disposal and the overall project cost.

7. The hydraulic dredge is the preferred equipment for most sediment removal operations. It is comparatively efficient in terms of production and relatively clean (resuspension of sediments) in its operating cycle. Special purpose dredges reduce resuspension of sediment, but are relatively inefficient in terms of energy expended to remove a given volume of sediment. Therefore, operational costs are usually higher than for conventional hydraulic dredges. Their use may be warranted only when toxic substances are being removed.

8. An average unit cost for sediment removal by dredging cannot be stated. The variables are so numerous that an average figure is meaningless. About the only generalized statement that can be made about unit cost is that it is likely to be inversely proportional to the amount of sediment removed.

9. Numerous uncertainties are associated with dredging as a lake restoration technique, e.g., the longevity of effectiveness, the relative effectiveness of partial compared to whole lake dredging, and the need to minimize ecosystem shock and encourage rapid repopulation of desirable organisms. Where nutrient control is of major interest, more experience is needed with surface sediment skimming compared to deepening as a restoration technique. Greater understanding of the comparative effectiveness of these two techniques could have a major influence on future projects. Costs could be reduced significantly if it were demonstrated that surface skimming of sediments could accomplish the same objectives as deepening. More sustained research and monitoring of diverse sediment removal methods is required.

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